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Limitations of the Current Practices Used to Perform Ecological Risk Assessment

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Special Section

ABSTRACT

The framework for ecological risk assessments has provided a way to analyze stressors in the environment. Despite the power of this tool to inform environmental management decisions, the practice has not reached its full potential. In this paper, limitations of the practice are described under 2 categories, namely inherent and contrived. Inherent limitations are constraints of nature that we need to be aware of as we design and interpret studies. Contrived limitations are impediments that have arisen in the practice through precedent or policy. The closing portion of this paper provides a series of short-term and long-term steps that could remove some of the limitations, especially the contrived ones, and improve the usefulness of risk assessments.

Keywords: Stochasticity Ecotheocracy Ecotoxicology Ecological scale

INTRODUCTION

Ecological risk assessment, as a formal entity, is relatively new in comparison to the many disciplines of science that feed into the process (Suter 2008). From initial efforts in the mid- to late 1980s, many sophisticated computational models and analytic methods have been developed. Whether simple or complex, risk assessments have been used effectively to inform decision-makers on a range of environmental management issues (Barnthouse 2008). Yet, the application of risk assessment as a science-based soft technology is not embraced uniformly as a useful tool (Shrader-Frechette 1991). Many environmentalists consider risk assessment to be a tool that affords industry and government the justification to do what they please. Indeed, during the so-called Gingrich revolution opponents of proposed changes to existing environmental laws included risk assessment along with takings and unfunded mandates as the "unholy trinity" (Paxman 1994). Even those who routinely use risk assessment to help make decisions find it wanting, as detailed in the review of the Coeur d'Alene River projects; a remarkable indictment of the environmental management process (NRC 2005).

Guidance for conducting risk assessments falls into 2 categories, namely broadly constructed frameworks (USEPA 1998 and predecessor documents) and more detailed technical directions (e.g., Wildlife Exposure Factors Handbook, USEPA 1993). Some of the specialized language of risk assessment (e.g., assessment endpoints, measurement endpoints) defined in early documents (such as Suter 1989) posed challenges to many practitioners. The Generic Ecological Assessment Endpoints report (USEPA 2003) was produced to alleviate misunderstandings and to illustrate applications of the different endpoint types. Periodically, new guidance documents and regulatory policy explanations were published to address ambiguities and to foster consistency. Nevertheless, among practitioners there remain many frustrations and much confusion regarding the application of risk assessment procedures on real projects.

My examination of problems encountered in the proper execution of risk assessment practices resulted in the identification of 14 significant limitations. I divide these limitations equally into 2 broad categories, namely those that are inherent to the science and those that are derived from behaviors of practitioners. The 2 lists are not exhaustive nor are they presented in order of importance. In general, the inherent limitations are those of any science-based endeavor; they should not be construed as impediments to moving forward, but rather recognized as part of the uncertainty that comes with the territory. Also, in general terms, limitations resulting from behaviors, here termed "contrived limitations," are ones that practitioners in the field have purposefully or otherwise imposed on the practice. Certain inherent limitations may be overcome through normal advances in the underlying sciences used in the ecological risk assessment process (e.g., ecology, toxicology, chemistry), but some will always remain limitations. Similarly, some contrived limitations could be eliminated relatively quickly by modifying the way risk assessments are performed or by adopting different policies, including ones that minimize the specter of dueling scientists that detracts from the management decisions to be made.

The issues identified in this paper are presented with the intent that practitioners and end users will gain a better understanding of what risk assessments can offer to the field of environmental management. The inherent limitations are ones that should be acknowledged honestly and should be reflected in our interpretations and recommendations. The contrived limitations are ones that we should strive to eliminate. If we accomplish both, risk assessments can become even more useful than they are currently.

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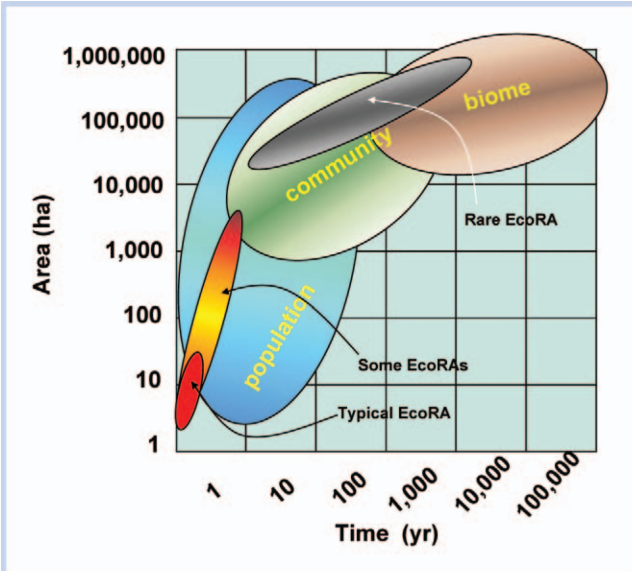

Figure 1. Relationship between spatial and temporal scales for ecological developments and risk assessments.

Inherent limitations

The 7 limitations discussed in this section are focused on the dynamic features embodied in ecology, and technical limitations in generating toxicity and exposure data that addresses the breadth and depth of biodiversity. Inherent limitations describe the platform upon which environmental management practices are carried out. These limitations also establish the boundaries of what is possible within the overlapping spheres of ecology, environmental management, and policy.

Stochasticity—May's (1976) examination of patterns in population dynamics moved ecological analysis into a new era. May's observations echoed contemporary discoveries in meteorology and fluid dynamics and launched a new way of thinking about nature (Gleick 1987). This new science of chaos demanded a different realm of predictions. Previously, scientists assumed that models could predict outcomes accurately through a series of equations. Errors in predictions were thought to be due only to imperfect knowledge, correctable by adding one more coefficient, one more term, or one more equation. The sea change in analytical thought caused by chaos theory is that we can no longer expect to predict the precise outcome of a population, a community, or an ecological system's functional processes; rather we need to express possible outcomes as probabilities.

Ecological systems are stochastic. Slight variations in initial conditions of a population and the magnitude of stressors, such as toxicants that alter survival or fecundity, acting on that population will result in several possible outcomes—predictions about each possible outcome are thus expressed as a probability. When evaluating predictions developed in the context of an ecological risk assessment, we might now conclude

- Case 1—Population A exposed to stress X is predicted as having an 80% chance of extirpation over 10 y;
- Case 2—Population B exposed to stress Y is predicted as having a 10% chance of extirpation over 10 y; and
- Case 3—Stress Z is predicted to have no adverse effect on population C.

As risk assessors, we are now faced with the reality that certain uncertainties are certain—we cannot know precisely how a particular population will respond. This introduces several challenges regarding communication of risk predictions, as well as being prepared for the wrath that is sure to come when the population responds against the odds. For example, what if in

- Case 1, after 20 y, population A is thriving;
- Case 2, within 1 y, population B is extirpated; and
- Case 3, population C declines 20% per y over 4 y?

Just as meteorologists express the likelihood of rain occurring in a specified area and a specified time, risk assessors can only express the likelihood that a particular outcome will occur. Similarly, in ecology, we need to communicate the boundaries of our predictions.

Scale (space, time)—Ecology, the study of the distribution and abundance of organisms and the way they interact with their environment, is all about scale, both in terms of space and time. There is a vast spread in the spatial scale required to support the diverse populations of living organisms. For some soil microbes the relevant scale to observe population dynamics or functional processes is in the realm of cubic millimeters to cubic centimeters, while in the case of the giant mycorrhizal fungus *Armillaria bulbosa* one individual organism can occupy several hundred hectares (Smith et al. 1992; botit.botany.wisc.edu/TOMS_FUNGI/apr2002.html). Thus, the scale for microbial organisms spans 10 orders of magnitude! For vertebrate populations, home ranges (excluding migratory species) span 4 or 5 orders of magnitude, with some exceeding 100000 ha (Carlsen et al. 2004). In contrast, most contaminated sites measure in the tens of hectares, and rarely exceed 5000 ha. Tannenbaum (2005) noted that for a 25-ha (50 acres) site there would be <1 to <3 resident individual organisms of the common terrestrial assessment species such as black-tailed jackrabbit, coyote, long-tail weasel, mule deer, raccoon, red fox, and white-tailed deer. Similarly for temporal scales, ecological processes for the most part occur over decades to tens or hundreds of millennia, but most risk assessments focus on a few years; the rare risk assessment forecasts results out to 100 y or more. There often is a considerable disconnect between the spatial and temporal scales relevant to ecological developments and the respective scales of investigations used in risk assessments (Figure 1).

Baseline ecological information—A common concern of environmental management and the focus of ecological risk assessments is the establishment of a reference baseline condition that can be used to evaluate pre- and postconditions for specified endpoints. The rationale for establishing a reference baseline carries some intriguing philosophical baggage. Implicit in the pursuit of the baseline is an assumption that there exists a stable ecological condition but for the actions of humans. In North America this is often thought of, either implicitly or explicitly, as a time prior to major European settlement. How frequently these discussions begin with some reference to pristine conditions. But as Krech (1999) established, Native American activities very measurably altered the North American landscape prior to European settlement. More fundamentally, with or without humans acting on the landscape, climate-driven ecological succession was occurring since the last glacial epoch waned some 10000 to 15000 y ago.

In large part, the search for a reference baseline reflects our collective desire to define the environmental conditions we

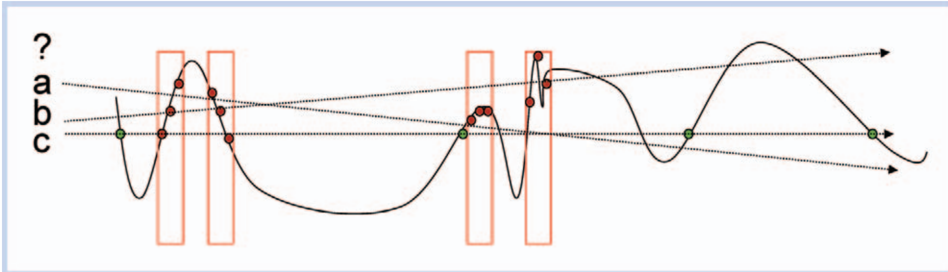

Figure 2. Stylized trajectories (**lines a, b, or c**) superimposed on an ecological condition (a population or a process such as productivity) over time with arbitrary sampling periods (red rectangles and red dots). (Line c [green dots] would imply constancy [stability], a condition that does not exist.)

wish to manage toward (Landis and McLaughlin 2000). However, the search for this elusive ecological baseline will be difficult. At best, we can describe a snapshot view, a fixed point in time, in which we characterize static conditions. Chaos theory shows the near-impossibility of establishing a clear trajectory of the vegetation and wildlife of interest at any historical moment. The one thing we can be certain about is that change was occurring. If some prior landscape condition is desired, then ecologists, it would seem, have an obligation to clearly describe those conditions that are possible and those that are unattainable.

Even if we agree on the description of desired baseline conditions, there remain many challenges in monitoring the changing status of those conditions. Referring to the earlier discussion of temporal scale, one challenge is to match the frequency of observation to the relevant scale at which ecological processes occur that is over decades, centuries, millennia, and beyond. The problem with only making observations over short time periods (seasons or a few years as is typical of risk assessments) is that short-term trajectories may give false indication of long-term trends. Observing a "fortuitous change" that coincides with, but is not fully consistent to one hypothesis can be misleading (Figure 2). Consider how remarkably different the conclusions might be among the 4 different sampling periods illustrated, and how such conclusions would alter environmental management decisions, if not viewed in the context of a longer term trajectory.

The challenge of tracking changes in ecological resources is addressed in part through adaptive management, a tool used by most resource agencies such as fish and game management departments, forestry, and water resource management units. However, adaptive management still is subject to uncertainties about baseline trajectory.

Toxicity profiles—Contemporary ecological risk assessments focus primarily on chemical effects and exposures. With respect to effects, and against the backdrop of biodiversity (Table 1), there remains a need to assume that species used in toxicity tests either as surrogates or indictor species reasonably reflect the response of other species of interest. Compiling responses across taxonomic groups has proven useful in addressing questions raised when extrapolating a given species' toxicity data to a species not yet tested, at least in terms of understanding the uncertainty inherent in such extrapolations (Figure 3). With the biodiversity of different taxonomic groups measured in the thousands to millions of species, but the number of tested species in each group tallied in the single digits to tens, much remains that we will never know.

Species sensitivity distributions have been used effectively to characterize the relative sensitivity among species tests (Postuma et al. 2000). Analysts have extrapolated from toxicity data in an species sensitivity distribution to derive an ambient concentration considered protective of a desired ecological condition. This approach has been most useful in aquatic toxicology due to the existence of a relatively large database from which to draw. For terrestrial taxa, species sensitivity distributions have so far proven less useful, due to the small number of test species used in definitive toxicity tests. Consequently, the uncertainty bounds on the tails of these distributions are generally large.

Exposure conditions—The other major component in the risk analysis portion of ecological risk assessment is exposure assessment. The USEPA's *Wildlife Exposure Factors Handbook* (1993) provides extensive summaries of data for use in estimating the various parameters that constitute exposure. These include dietary preferences, seasonal foraging ranges, ingestion rates for food and water, metabolic (caloric) demand for some taxa, body sizes by age and gender, as well as incidental ingestion rates of soil or sediment for several taxa. As extensive as these data are, there remain several important physiological and ecological features for which few if any data exist.

Features such as behavioral dynamics, metabolic requirements, and interactions with nonchemical stressors, which can greatly alter foraging patterns and exposure (Hope 2004), generally are ignored in risk assessments. Uncertainty with respect to diet is seldom described and rarely quantified.

Table 1. Species richness by major groups (adapted from EPT 1996)

Mammals	**~4000**	**Conifers**	**~550**
Birds	~8600	Ferns	~13000
Fish	~40000	Algae	~28000
Invertebrates	~10000000	Bryophytes	~32000
		Fungi	~100000
		Angiosperms	~235000

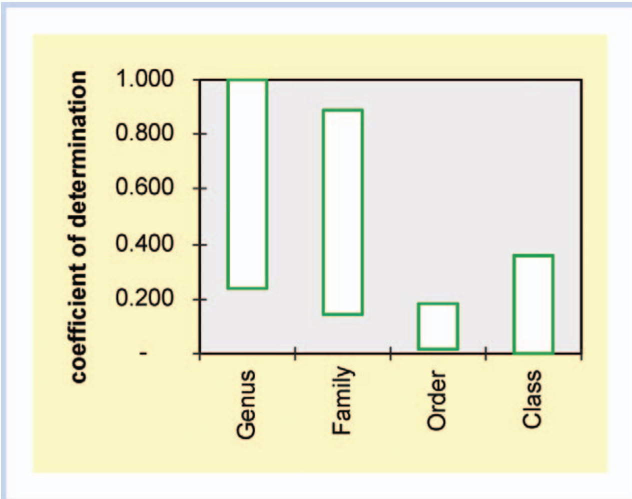

Figure 3. Coefficient of determination for plant toxicity tests within taxonomic hierarchies (adapted from Fletcher et al. 1988; EPT 1996).

Finally, there typically is little if any quantitative information regarding the bioavailable fraction that passes from food items or soil/sediment into the blood stream. It is reasonable to conclude that what we don't know about exposure exceeds what we do know!

Multiple stressors—Arguably, no organism resides at the optimum position for all of its niche parameters (Figure 4). In other words "stress" is a constant. However, physiological mechanisms provide organisms with the means of finessing the effects of specific stressors through the adjustment or realignment of their baseline optimal conditions. For example, as the weather warms from spring through fall, plants effectively shift their response to temperature gradually adjusting to warmer conditions and then reversing this trend in the fall. In northern temperate climates, temperatures readily tolerated by plants in April or May can kill those same individuals should those same temperatures occur in July or August. Anticipation and acclimation are important survival mechanisms for organisms.

When environmental conditions shift toward the edges of tolerance for individual organisms or populations, genetic changes can occur either through mutation, activation of "silent" genes at the organism level, or changes in gene frequency at the population level. Genetic shifts of this nature, referred to as adaptation, demonstrate important evolutionary mechanisms that adjust an organism's, and ultimately a population's, fitness to changing environments. With multiple stressors present, the cumulative effect of sequential adaptations will confound predictive capacity regarding one particular stressor's effects. Excellent discussions regarding the challenges of dealing with multiple stressors appear in Foran and Ferenc (1999) and Ferenc and Foran (2000).

Complex stressors—Complex stressors are those which cause different effects under different circumstances (Dorward-King et al. 2001). Examples of this include differential responses to essential nutrients across the range of concentrations from deficiency through sufficiency and finally toxicity. Also with essential nutrients, there are differential responses to a given nutrient depending upon the co-occurrence of paired nutrients (e.g., copper and molybdenum). Similarly, response to a stressor depends on the degree to which the exposed organisms are acclimated or adapted to the particular stressor. Most interesting are the situations where the sequence of exposure to different stressors results in different ecosystem level responses (de Ruiter et al. 2001). At the organismal level, the magnitude of the effects of disease is often seen in the context of prior conditions (Figure 5), where the cumulative effects of several precursor stresses predisposes the individual to a more pronounced response (e.g., in elderly humans the broken hip followed by pneumonia, the elk herd bull succumbing to infection following the rut).

The implications of responses to a set of complex stressors in ecological risk assessment can be quite profound. Though some of the better studied relationships such as elemental pairs copper:molybdenum or zinc:cadmium, as well as pH:ammonium are often considered, responses to complex stressors, if acknowledged at all, are seldom incorporated into risk assessments. Other than the US Fish and Wildlife (USFWS 1997) protocol for testing to demonstrate acceptability of nontoxic shot, in which the test animals are raised on nutrient poor conditions designed to simulate winter migration stress conditions (Brewer et al. 2003), most test protocols demand that test organisms are at their healthiest. Assessing the implications of exposure to a set of complex stressors in a risk assessment may be most important for forensic investigations (also known as retrospective risk assessments). However, when monitoring an ecological system as a means of evaluating the predicted consequences of a release, complex stressor interactions could be highly significant. Examples of guidance that acknowledges complexities of stressor interactions are the Metals Environment Risk Assessment Guidance (2007) and PAHs (Douben 2003).

Contrived limitations

Limitations derived from behaviors of practitioners are also introduced under 7 topics. The section begins with a discussion of ecotheocracy, the application of beliefs and normative

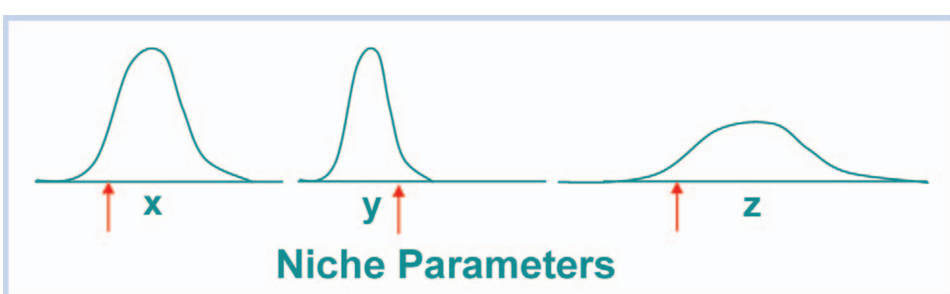

Figure 4. Generalized illustrations of an organism's physiological tolerance to 3 environmental variables (the red arrow, indicating a current condition outside of the optimal conditions).

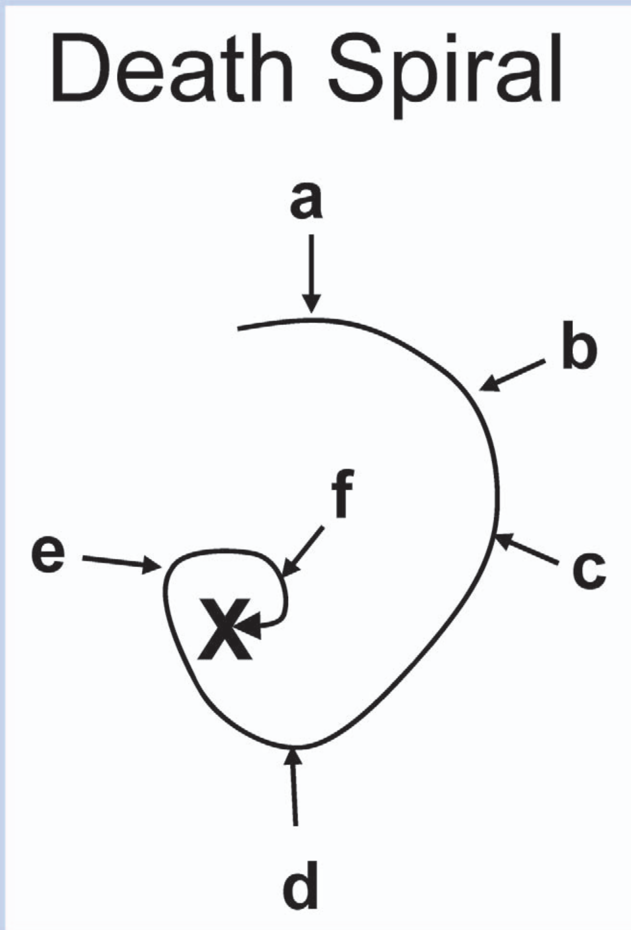

Figure 5. Depiction of the gradual weakening or declining fitness of an organism from a series of stressors such that the proximal event preceding death might otherwise have been innocuous—the so-called death spiral.

science, as a serious impediment to risk assessment and ultimately environmental management. This is followed by a brief discussion about limitations imposed through legal or regulatory requirements, and limitations resulting from policies promulgated by agencies to carry out legal mandates, including one that deserves special attention, namely the continued use of point estimates in the risk assessment process. Data quality and the reporting of relevant data are identified as serious limitations on the use of existing toxicological data. Perceived costs associated with conducting quality ecological risk assessments, and trustworthiness of the process and practitioners, are identified as the final 2 limitations.

Ecotheocracy—Kapustka and Landis (1998), in arguing that ecology, as a science, is value-neutral, described the disconnect between ecology as science and ecology as a belief system that permeates much of the environmental movement. They attributed some of the catechism of ecotheocracy, (i.e., goodness of nature, integrity, stability, balance of nature, recovery, and restoration) as being derived from Clementsian views. Two more recent additions, ecosystem health and grand design, are outgrowths of this same belief system. Norton (1996) examines the philosophical penchant for embracing constancy dating back to Plato, embedded into Judeo-Christian theology, and carried forward to contemporary environmental passions from musings of John Muir through Fredric Clements to Eugene Odum. Though the basic concepts presented by Clements (1916) were effectively refuted by Gleason (1926), Clements' views dominated ecology well into the 1960s. It is not surprising that the modern environmental movement, begun in the late 1960s and early 1970s, adopted themes that emphasized the goodness of nature and the closely coupled view of the evil of humankind.

Metaphors relating ecological entities with organismal features, especially mammalian organs such as wetlands being the kidneys of a watershed, have been used to try to explain complex features. Perturbations of ecological systems are often presented as analogous to human health situations such as the accumulation of litter makes a forest unhealthy, but Suter (1993) cautioned against this as ecosystems "are not analogous to the patient in medicine." Lackey (2001) examined arguments for and against using the metaphor "ecosystem health" in detail, drawing upon an extensive body of literature to illustrate the implications to policy should ecosystem health be adopted as a goal in ecological policy. Without taking a stand either for or against the notion of ecosystem health, Lackey concluded, "Whether or not one finds intellectual sustenance in the notion … the policy concerns its proponents attempt to confront are genuine."

Hierarchy theory shows that one can ask the same ecosystem questions about an organism or a landscape (that is, what quantity of solids/liquids/air went in, what quantity came out, how did biomass and temperature change, etc.). So, while one can't compare ecosystems to organisms directly, similar questions may be approached by explicitly recognizing a change in scale and type. Where ecotheocracy steps out of ecology as a science and into desired belief is in a failure to recognize this change in scale (Allen and Hoekstra 1992).

Lackey (2007) rejects the assertion that science is value-neutral, but cautions against "overstepping our role as scientists and slipping into stealth policy advocacy" of normative science because "scientific information is too important to the resolution of vital, divisive, and controversial ecological issues to allow scientists to marginalize science through its misuse." The Kapustka and Landis (1998) perspective holds that assessment endpoints based on Clementsian values (listed above that are belief-based rather than science-based) are unacceptable as science-based assessment endpoints and inappropriate as measurement endpoints. In short, it is indefensible to use them and then claim a science-based effort.

Legal or regulatory—Unfortunately, several environmental laws (e.g., the Endangered Species Act, Clean Water Act in the United States) echo the themes of Clementsian ecology as they reference "the balance of nature" (Profeta 1996). To the extent that some laws specifically or implicitly address such themes, they can impose limitations on applications of contemporary science. Public laws reflect the values of society, at least as interpreted through the legislators authorized to enact them. Justification for performing ecological risk assessments in one form or another is derived from the enabling language of specific laws and the regulations that Agencies promulgate to carry out the laws (see USEPA 2003 Appendix A for specific linkages). Beyond the justification, the language can be interpreted as dictating certain approaches or boundaries of the assessment process. At the state level, for example, Oregon (Oregon DE [Oregon DEQ] 1998) explicitly calls for probabilistic assessments.

In some cases, such as the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), also known as Superfund, the potential liability associated with identifying certain effects, either in terms of remediation costs or natural resource damages, can have unintended consequences that make it better "not to know," or to only do the minimum required to characterize risks—large uncertainties provide a certain level of legal cover. Prescriptive measures also can stifle innovation or at least provide justification for using minimalist approaches.

Policy or precedent—There is a long list of practices stemming from policy or precedent that severely limit the quality of risk assessments. The most prominent of these are the continued reliance on the no observed adverse effect concentration (NOAEC), the lowest observed adverse effect concentration (LOAEC), and the maximum acceptable toxicant concentration (MATC) calculated as the geometric mean of the NOAEC and LOAEC. These endpoints are constrained by limitations of the analysis of variance (ANOVA) experimental design used to generate the values. The ANOVA design has been criticized for being dependent on the concentration intervals chosen, for being insensitive because of inherent variability in responses, and for the fact that most of the information from the test is lost. The technical community has reached consensus with the arguments regarding the problems of using "no effect" determinations based on ANOVA designs (Hoeskstra and van Ewijk 1993; Laskowski 1995; Chapman et al. 1996; Bailer and Oris 1997; OEC [Oregon DEQ] 1998). Indeed, understanding of these terms (i.e., NOAEC and LOAEC) was so diffuse that the ASTM E47 Terminology Sub-Committee required nearly 8 y and some 15 ballots before consensus was reached; it passed primarily because the terms continue to be used to comply with policies and precedent.

A 2nd policy-driven precedent constraining the value of risk assessments is extensive reliance on hazard (or risk) quotients, when other tools could be used (Sorensen et al. 2004). These unitless derivations are merely a comparison of a measured or predicted environmental concentration in a medium of interest to a toxicity endpoint. Often the endpoint of choice is some expression of a threshold over which there is some indication of adverse effects. Because these are unitless values, there apparently is great temptation to add the quotients, a practice that is easily abused. The input of toxicity information in the quotient is limited to the threshold value; all of the information about the slope of the toxicity response is discarded. As a consequence, the magnitude of the quotient has no meaning when comparing among substances. For example, due to the very gradual slope for lead toxicity, a 10-fold exceedance of a threshold value may translate to a minor effect on receptors, whereas a 2-fold exceedance of the selenium threshold may likely be lethal to most organisms. The quotients can be highly useful as a screening tool to eliminate some substances, media, or receptors from further consideration, but it would be best to limit the use of quotients to make simple determinations of "in" or "out" of consideration for subsequent risk iterations.

However, all too many risk assessors spend great effort tweaking the assumptions to influence the magnitude of the quotient. On the "protectionist" side of the debate, there is much energy devoted toward inflating the exposure estimates and deflating the threshold values. Thus a host of "safety factors" (also known as assessment factors, uncertainty factors) are applied. The regulated community, on the other hand, strives to do the reverse (that is, to minimize the estimates of exposure through application of bioavailability fractions, area-use factors, and other modifiers). These adversarial tensions lead to unfortunate situations where predictions of dire consequences are made, but there is lush and diverse vegetation and robust wildlife populations. On the flip side, moonscapes are depicted as areas in transition with no toxicant causalities. We ought to leave this entertaining exercise of "adding or subtracting angels on the heads of pins," and move toward the more appropriate applications of higher-tiered risk assessment practices.

In developing risk assessment procedures, the USEPA (1993) compiled relevant information on a number of species commonly used in ecological risk assessment in its *Wildlife Exposure Factors Handbook*. Though the descriptions included aspects of life histories of the primary target species and various secondary species, the information on habitat requirements was cursory at best. At about the same time as the *Wildlife Exposure Factors Handbook* was prepared, the US Fish and Wildlife Service (USFWS 1981; Schroeder and Haire 1993) published formal descriptions of wildlife habitat requirements in the form of habitat suitability index models. Ironically, there are only a few species in common in the lists from the *Wildlife Exposure Factors Handbook* and the habitat suitability index models (Kapustka et al. 2001); how much better ecological risk assessments could be if these efforts had been coordinated to focus on the same species.

Point estimates—Experimental design can have a profound influence on toxicity test values derived from the resulting data. The overwhelming majority of toxicity tests have used an ANOVA design as described above. Stephenson et al. (2000) and van Assche et al. (2002) argued for the use of regression-based study designs with unequal number of replicates spread over 10 or more concentrations focused around the putative threshold concentration. From this, if one uses a point estimate to generate a quotient for use in screening, other information contained in the typically nonlinear concentration- or dose-response relationship remains available to explore the magnitude of expected results along the concentration gradient. Based on the preponderance of arguments against use of NOAECs or LOAECs, and the availability of a superior experimental design, the continued use of these point estimates is indefensible in a process claimed to be grounded in the sciences.

Data quality—The often low quality of data available for use in risk assessments is troubling. This conclusion was driven home most prominently in the USEPA Ecological Soil Screening Level (Eco-SSL) program (www.epa.gov/ecotox/ecoss/index.html). After agreeing on the minimum reporting requirements to qualify a study for use in the Eco-SSL process, the work groups repeatedly encountered fatal gaps in reporting test conditions, procedures used, and completeness that rendered the data unusable for derivation of Eco-SSLs. Commonly, the limitations appeared to be a consequence of page limits or limits imposed by the editors on the number of tables and figures, resulting in critical information, surely gathered in the study, not being available for independent verification of endpoint derivations. Such verification was deemed essential, as quite often data presented in tables or figures were at odds with statements in the text. For example, tabular or graphic presentations with statements of no statistical differences would be followed by conclusions such as "though no statistical differences were found, the NOEC was X." Other reports would show trend lines of means with no accompanying indication of variances or other measure of statistical significance. Of the large number of reports on any

Table 2. Tally of references examined pertaining to PAHs for 5 receptor groups[a]

Receptor group	ProCite® hits	Surviving screen and ordered	Acquired[b]	Passed acceptance criteria
Plants	496	162	91	1
Invertebrates	182	81	68	3
Birds	509	76	40 (35)	1
Mammals	4122	283	178 (139)[c]	9
Herpetofauna	197	28	27 (21)[d]	0
Retention rate		11.40%	3.20%	

[a] Unpublished technical report prepared by Ecological Planning and Toxicology, Corvallis, OR, USA under contract to the American Petroleum Institute.
[b] The numbers in parentheses are the numbers actually reviewed if fewer than acquisition number.
[c] Distributed as follows: rats, 71; mice, 46; other (guinea pigs, rabbits, hamsters, etc.), 22.
[d] Distributed as follows: amphibians, 18; reptiles, 3.

receptor group–chemical combination, that at first would seem to have useful information, only a handful can actually be used in setting screening values (see Tables 2 and 3). Unfortunately, most of the seemingly relevant peer-reviewed literature lacks the documentation to be useful in the risk assessment screening process. In that so much of the literature isn't suitable for screening, it begs the question of whether the data are useful in any portion of the risk assessment.

Limited investment in toxicity testing by government institutions and the private sector, and the curtailing of tests in deference to animal rights concerns, make it doubtful that new, substantive in vivo testing will be available to correct obvious gaps in the toxicity literature. The taxonomic diversity of terrestrial test species is very low compared to that for aquatic systems. Therefore, we can continue to expect large uncertainties in toxicity information needed for ecological risk assessment.

Perceived value and perceived cost—The connection between ecological risk assessments and decisions made by environmental managers is often obscure or lacking (see NRC 2005). In many circles, risk assessments, like environmental impact assessments, are viewed as regulatory requirements that must be met, not as something that informs the decision process. Indeed, the oft-used phrase, "to support the environmental management decision" is viewed by some as selectively sifting information to justify a decision already made. To the extent that risk assessments are seen as required exercises that may not add value to a project they represent missed opportunities to inform meaningful management strategies. There also seems to be a common failure to match the level of effort to the magnitude of problems being addressed. As the perceived value rises, the costs of doing high quality risk assessments shift from being an unfortunate cost of doing business to an investment needed to make informed decisions. Indeed, the cost of making a poorly informed decision can have serious unintended consequences as well as being costly monetarily. For example, a predetermined remedy to remove contaminated sediment from a wetland or floodplain may not be warranted if risk were

Table 3. Tally of references examined pertaining to chemicals of concern for plants and invertebrates[a]

Receptor group	ProCite® hits	Surviving initial screen	Papers scored	Experiments scored	Useful data
Antimony-plant	300	58	0	0	0
Barium-plant	622	60	1	2	1
Chromium (VI)-invertebrate	1485	121	1	1	0
Chromium (VI)-plant	2679	366	3	12	2
HMX-invertebrate	40	22	3	9	2
HMX-plant	43	9	0	0	0
Nickel-invertebrate	1534	94	2	7	4
Perchlorate-invertebrate	145	6	0	0	0
Perchlorate-plant	91	9	0	0	0
Silver-invertebrate	749	50	0	0	0
Silver-plant	1350	291	1	12	4
Totals	9038	1086	11	43	13
Retention rate (% of hits)		12%	0.12%	—	—

[a] Unpublished technical report prepared by Ecological Planning and Toxicology, Corvallis, OR, USA under contract to the US Navy. HMX = cyclotetramethylene-tetranitramine, tetrahexamine tetranitramine. — = no data.

assessed. Not only would removal effectively eliminate the habitat of the species intended to be protected, but the intrusive actions are expensive. Examination of costs of ecological risk assessments and savings due to targeted remediation range from >\$50 to >\$500 for each dollar spent on risk assessment (L. Kapustka and D. Nikl, Golder Associates, North Vancouver, BC, Canada, unpublished data).

Trustworthiness—Many critics of risk assessments distrust the entire process. The perception that risk assessment is easily gamed (that is, data are manipulated until the "risk" can be whisked away) is fueled by some of the extraordinary efforts that go into whittling down exposure estimates and bumping up threshold responses (as discussed above under the risk quotient limitation). Also, the perception that irrevocable decisions are made prior to characterizing risk (e.g., the siting of a facility determined and then a set of alternative sites chosen for the evaluation; the remediation option selected before the risk assessment is performed) adds to the level of cynicism. Finally, poor quality risk communication fans the flames of discontent about risk assessments, further undermining the utility of risk assessment.

In the interplay of policy and regulatory action, there inevitably are varying degrees of tension that arise due to differences in stakeholders' tolerance or acceptance of environmental risk. Creation of these tensions often follows directly from the processes used in reaching decisions, but there is much more. Explorations into risk perception from nearly 2 decades ago have provided powerful illustrations into the way people handle multiple forms of information as they make decisions (Covello et al. 1988). In general, they tell us that scientific or technical descriptions of a risk event or activity form only a small part of the information that people process as they consider accepting or rejecting the risk.

Trust in any arena has to be earned. The risk assessment community, by and large, has performed poorly in terms of laying the groundwork for earning trust. The shortage of formal follow-up studies to evaluate risk predictions seems contrary to accepted science practice and is in conflict with the frequent portrayals that risk assessments are science-based. Without monitoring to corroborate risk predictions and then "calibration" of the procedures to improve the predictive capacity of risk assessments, trustworthiness will remain a major limitation of ecological risk assessments.

Path forward

After acknowledging limitations, it is important to consider what, if anything might be done to overcome them. The suggestions that follow are divided arbitrarily into near-term and long-term programs. Many of the contrived limitations, because they are creations of humankind, can be addressed if there is the political will to do so. Some at least conceptually could be fixed easily; others will require much effort over long periods. The inherent limitations will be easier to live with when we explicitly acknowledge their existence and actively seek a higher degree of understanding.

Near term (<3 y)—In the near term, as practitioners and consumers of risk assessments, we should strive to align our policies and practices with the state of the science. Examples of this include

- Use ECx (not NOAEC or LOAEC) for screening and use the complete toxicity response profile for higher-tiered assessments;
- Restrict use of risk quotients to screening-level assessments and use effects response relationships for higher-tiered assessments;
- Adopt contemporary ecological theory and practices in defining assessment endpoints, conducting analysis steps, interpreting consequences, and proposing risk mitigation/reduction actions;
- Establish focused follow-up activities to evaluate risk predictions including monitoring for corroboration or calibration of risk assessment procedures;
- Promote integration of risk assessments with environmental management decision processes; and
- Initiate long-range research programs to tackle the long-term issues.

Long term (3+ y)—Long-term activities should focus on filling data gaps and moving forward on a number of initiatives already explored in professional circles and within the risk assessment industry. Candidate topics for consideration include

- Increase the focus of risk assessments on functional ecological processes at population and community levels;
- Expand the scope of risk assessments to include biological and physical stressors explicitly (i.e., put chemical stressors in ecological context);
- Adopt landscape-perspective assessments and configure necessary regulations and policies to approach meaningful ecological scales, including increasing the use of meta-analysis to aggregate ecological effects information at ecoregional levels;
- Develop the data needs for proper terrestrial wildlife species sensitivity distribution analyses, comparable to that being done for aquatic receptors; and
- Evaluate risk predictions with analyses contained in State of Environment Reports produced by governments and by various nongovernment institutions.

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